LAWRENCE
LIVERMORE
NATIONAL
LABORATORY

UCRL-TR-201702

Technical Basis for Also Using Health-Risk Assessment to Establish Contaminant Boundaries for Corrective Action Units (CAUs) of the Underground Test Area (UGTA) at the Nevada Test Site (NTS)

J.I. Daniels & A.F.B. Tompson

November 25, 2003

DISCLAIMER

Work performed under the auspices of the U. S. Department of Energy by the University of California, Lawrence Livermore National Laboratory under contract W-7405-ENG-48.

Contents

Introduction 1
Definition of Contaminant Boundary 5
Bases for a Contaminant Boundary 7
Risk-Based Boundary 9
The MCL-Based Boundary and Conformity with Health Protection 11
Fostering Risk Communication to Educate, Inform, Enlighten, and Fully Disclose the Goals and Structure of Contaminant Boundaries for CAUs at the NTS 17
Conclusion 19
Acknowledgments 20
References 21
Appendix A: Radiation, Radioactivity, Related Dose and Its Special Units, and Risk Coefficients and Their Applications 23

Technical Basis for Also Using Health-Risk Assessment to Establish Contaminant Boundaries for Corrective Action Units (CAUs) of the Underground Test Area (UGTA) at the Nevada Test Site (NTS)

Introduction

The locations of underground nuclear tests conducted at the Nevada Test Site (NTS) are grouped into geographically distinct corrective action units (CAUs) as part of the corrective action strategy for the Underground Test Area (UGTA) Project. Within each CAU, the corrective action objective is to utilize hydrogeologic, geochemical, and radionuclide inventory data to develop a computer model of groundwater flow and radionuclide transport away from each test location. Computer simulations will be used to estimate over a 1,000-y time period (or at any selected time[s] up to 1,000 y), with uncertainty quantified, the greatest vertical and horizontal extent of radionuclide migration in groundwater in each CAU, and the closed boundary beyond which levels of such contamination are unlikely to exceed those deemed protective of public health. Specifically, a contaminant boundary is the model-predicted perimeter (projected two-dimensionally at the surface from the three-dimensional spatial configuration obtained from modeling that includes the deepest affected hydrostratigraphic unit) at or inside of which radionuclide-contaminated groundwater exceeds levels considered protective of public health (e.g., an enforceable regulatory limit, such as standards defined by the Safe Drinking Water Act). Consequently, groundwater outside or beyond this contaminant boundary (at depth[s] and at time[s] over 1,000 y) is considered to be safe for domestic and municipal uses, especially when there is a predetermined high level of confidence associated with its predicted occurrence (e.g., ≥95% is a conservative level of confidence that is used for purposes of this discussion, but another value may be applicable and agreed upon in practice). This strategy is described in detail in Section 3.2 of Revision 1 to Appendix VI of the *Federal Facility Agreement and Consent Order* (FFACO, 1996; as amended), which in its entirety is applicable to all land in the State of Nevada controlled, managed, owned, or leased by the United States Department of Energy (DOE).

This technical basis document serves two purposes. First, it provides a detailed discussion of the rationale and procedures suitable for deriving a risk-based contaminant boundary that will protect public health unambiguously, along with examples that are intended as illustrative only to facilitate understanding. Second, it explains the benefits of using such information as the framework for fostering risk communication to educate, inform, and enlighten, and importantly, to fully disclose the goals and structure of contaminant boundaries.

To determine a contaminant boundary within a CAU, standards or criteria must be adopted that establish whether groundwater is safe or unsafe for public (and worker) use. For purposes of this discussion, drinking water consumption is considered the pathway of exposure. However, in practice, a realistic land-use scenario must be described and agreed upon before a prospective, realistic risk-based calculation is performed. Otherwise, it will not be clear whether consumption of drinking water is a even appropriate. For example, the future land use that is defined may not even permit access to the contaminated water (e.g., denial of use by law and stewardship; or a lack of accessibility), and in that situation there would be no exposure and no potential health consequences.

Taking into consideration that consumption of the groundwater is feasible, the groundwater is deemed unsafe if it contains radionuclide contamination that exceeds the criteria that the United States Environmental Protection Agency (EPA) considers in establishing regulatory standards for drinking water contaminants, including radionuclides *generally* (i.e., a target range for lifetime excess-cancer risk that is not to exceed 10^{-4} [1/10,000] and ideally is less than 10^{-6} [1/1,000,000]) (see EPA, 2000a). Thus, the maximum contaminant levels (MCLs) for substances in drinking water are considered to be health-protective and generally are derived on the basis of acceptable level of risk. Revision 1 to Appendix VI, Section 3 of the *Federal Facility Agreement and Consent Order* (FFACO, 1996; as amended) even stipulates that the contaminant boundary for a CAU is to be determined by modeling groundwater flow and radionuclide transport over the next 1,000 y, with uncertainty considered. Such modeling will identify over the entire 1,000-y period the spatial extent of radionuclide-contaminated groundwater, created by nuclear tests, that is above background conditions and exceeds the National Primary Drinking Water Regulations (NPDWRs) for radionuclides. These NPDWRs were developed as a requirement of the Safe Drinking Water Act (SDWA) and have been referred to as SDWA standards or as MCLs for radionuclides.

The current NPDWRs for radionuclides specifically, excluding uranium, were adopted originally in 1976 and readopted in 2000 and remain derived using outdated data and procedures (EPA, 2000a). For example, these NPDWRs place an emphasis on monitoring for the MCLs and/or assessing maximum limits on annual dose, depending on the nuclear-emission category for the radionuclide of interest (e.g., alpha particle [α] *vs.* beta [β] and photons [such as gamma-rays, γ]). Notwithstanding, a more unifying and realistic concept is to compute the lifetime-excess cancer health risk for an individual radionuclide, based on the activity concentration (Bq/L) of the radionuclide and consideration of exposure pathway(s), regardless of nuclear-emission category, and then sum risks when multiple radionuclides are involved. Interestingly, careful examination of the published results of EPA application of modern health-risk coefficients with current MCLs for specific radionuclides for the ingestion pathway revealed that some current MCLs for radionuclides are at levels that would conflict with EPA's own stated risk-based criteria for establishing regulatory standards for drinking water contaminants (see EPA, 2000b; EPA/USGS, 2000).

Similarly, application of modern health-risk analyses to the NPDWRs for radionuclides indicate unequivocally that when there are multiple radionuclides in different emission categories present in water, there can be instances when an MCL or dose limit can be met for a particular category of radionuclide, but the corresponding total lifetime excess-cancer risk for all radionuclides, regardless of category, can exceed the target-risk range considered to be health-protective. This dilemma occurs primarily because the current NPDWRs for radionuclides were derived *without* any consideration given to the unique situations where many different radionuclides are introduced into the subsurface environment by human actions. Therefore, even though the current NPDWRs for radionuclides represent the enforceable regulatory standards, in situations involving multiple radionuclide contamination in groundwater, the categorical MCLs (i.e., for gross alpha [excluding uranium and radon], gross beta and photon emissions, and uranium) alone are not always sufficient for defining the safe or unsafe level of radionuclide contamination in a potential drinking water supply. However, it is important to note again that in practice, it is the land-use scenario that will determine what, if any, exposures to the contaminated groundwater are feasible (or are prevented).

One solution to the described dilemma is to derive a CAU contaminant boundary consistent with an applicable lifetime excess-cancer-risk based criteria. An alternate approach, more inclusive of the language of the FFACO (1996; as amended), would be to compute both a risk-based and MCL-based boundary and compare them to derive a final contaminant boundary. Even though the language of the FFACO (1996; as amended) indicates that the contaminant boundary should be derived using MCLs, a risk-based approach better serves the mutual goals of the DOE and the State of Nevada Division of Environmental Protection (NDEP) to protect public health and safety *unambiguously*.

A procedure has been adopted and is in place for identifying all of the relevant radionuclides of concern for the source term used in CAU groundwater-flow and contaminant-transport modeling (see Appendix A in Pawloski et al., 2001), and that procedure is not discussed here. The CAU hydrologic-flow and contaminant-transport modeling and other geophysical-process modeling also are considered sufficiently robust to address the complexities of subsurface explosion phenomenology, hydrology, geology, geochemistry, and contaminant-transport parameters, with uncertainty considered. The hydrologic-flow and contaminant-transport models also are suitable for computing estimates of risk from model outputs of activity concentrations using additional equations addressing subsequent exposure and resulting lifetime excess-cancer risk. Although uncertainty can and should be addressed quantitatively with respect to the drinking water ingestion rate and exposure duration (see Daniels et al., 2000) required for computing lifetime excess-cancer risk, the oversimplified, well established and accepted regulatory practice of using *conservative* values for such parameters of 2 L/d and 365 d/y, respectively, for a 70-y lifetime are employed as an example for purposes of this discussion. For further information on the relationship between radiation, radioactivity, related dose and its special units, and risk coefficients and their applications with respect to establishing a contaminant boundary, refer to Appendix A.

Definition of Contaminant Boundary

The simplest way to describe a contaminant boundary is to first envision a closed geometric surface in the saturated zone of a CAU that separates the area where radionuclide contamination does not exceed a standard from the area where such contamination does exceed the standard. This surface can be considered:

- A **moving** boundary, if it is based upon instantaneous "snapshots" in time where locations in the CAU at which the standard is met or violated are identified, or
- A **fixed** boundary, if it is based upon an examination over an extended or historical period of time, such as 1,000 years, as to where the standard has ever been met or violated in the CAU.

In the current context, we may speak of using a "risk-based" standard, or an "MCL-based" standard to define this boundary, each of which will be discussed in more detail below. Regardless of which standard is proposed for use, the **fixed** boundary can be considered more conservative in the sense that it will envelop all volumes of geologic material and the contaminated groundwater that it hosts over the entire period of time considered by all simulations addressing the **moving** boundary.

Within the UGTA project, a discrete hydrogeologic transport model will be used to forecast the movement of radionuclides in groundwater through each CAU for 1,000 years. Such models will typically utilize three-dimensional simulation domains that include the entire geographic area of each CAU, or possibly a larger area if needed for specification of model boundary conditions or other purposes, and will extend from the water table to a suitable lower hydrostratigraphic boundary. The transport model will predict—as its main result—the spatial distribution of radionuclide concentrations, $c_i(\mathbf{x}, t)$, as a function of time, t, for all radionuclides, i, at all discrete locations $\mathbf{x} = (x, y, z)$ in the saturated zone of the CAU model domain for discrete times lying in the interval $0 < t \leq 1{,}000$ y. The concentration solutions will be used to evaluate whether a standard is met or violated at each discrete location **x**, either instantaneously or in a historical sense, and in turn, determine the location of the contaminant boundary.

It should be noted that the location of the model-predicted contaminant boundary—a closed three-dimensional surface underground—can be projected up to the ground surface to

define a corresponding two-dimensional contaminant perimeter boundary. This boundary will encircle (*x, y*) locations below which (in the saturated zone) standards are violated, but will not convey exact depth (*z*) information at these locations where such violations occur.

Now, because of uncertainties in the actual model application and the way in which computer models are developed, there will not be a single simulation with one **certain** solution, but, rather a multiple number of simulations that produce a correspondingly multiple number of alternative solutions (that are used collectively to address that uncertainty associated with any one, individual solution). Accordingly, there will be an uncertainty in the predicted location of the contaminant boundary, regardless of whether it is determined in a **moving** or **fixed** sense.

The suite or ensemble of solutions for $c_i(\mathbf{x}, t)$ at each discrete location **x** can be visualized as a two-dimensional matrix of concentration results, where rows represent realizations (n = 1 to N) and columns represent discrete times (t = 1 to 1,000 y). In addition to populating each "entry" with specific concentration data, "derived" results can be included to indicate whether conditions at that point (**x**), at that time (t) or historically over time (0 to t), and in that specific realization (n) satisfy or violate either the MCL- or the risk-based standards.

In terms of quantifying the degree of confidence for a specific contaminant boundary, we want to determine the locus of discrete points **x** (or model grid blocks) in which a reasonably high number of simulations (n) do not produce results or outcomes that exceed or violate one of the proposed standards, either at a particular time (t), or at any time over the lifetime of the simulations (0 to t). If this "high number" is sufficiently large, then we will conclude that this location lies **outside** of the contaminant boundary with a high degree of confidence, either for the particular time considered, or for the entire history covered by the simulations.

By "sufficiently large," for example, we could mean that 95% or more of the simulations yield an outcome that does not violate the standard of interest, which, in turn, implies that only 5% or less of them at this location would exceed it. Conversely, if fewer than 95% of the simulations yield an outcome at **x** that do not violate the standard of interest, then **x** is considered to lie **within** the contaminant boundary. In this case, more than 5% of the simulations would produce outcomes that exceed the standard considered within the boundary. Of course, other confidence levels (e.g., 50%, 60%, 80%, etc.) may be used to convey the concept of "sufficiently large."

Bases for a Contaminant Boundary

Among the plausible bases for describing a contaminant boundary, two can be considered:

- That boundary beyond which radionuclide contamination is unlikely to produce an unacceptable level of risk, either at a particular time (t), or over the entire 1,000-y evaluation period (i.e., the perimeter at or inside of which risk may exceed an acceptable level), or
- That boundary beyond which radionuclide contaminants are unlikely to exceed their respective MCLs, either at a particular time (t), or over the entire 1,000-y evaluation period (i.e., the perimeter at or inside of which the MCL for one or more radionuclide(s) may be exceeded).

Furthermore, two approaches for expressing the degree of confidence in the predicted location of the contaminant boundary can be considered:

- Quantifying the confidence level (e.g., ≥95% is a conservative level of confidence that is used for purposes of this discussion, but another value may be applicable and agreed upon in practice; see FFACO, 1996; as amended) in the estimate of a contaminant boundary based on the locations of ground water where the acceptable level of risk or the MCL(s) are predicted not to be exceeded at anytime over 1,000 y,

or, alternatively, as recommended by the NRC (1995) in a document describing *Technical Bases for Yucca Mountain Standards,*

- Determining if the expected value (mean) of the probabilistic distribution of health effects for members of the critical group exposed to a predicted level of radionuclide contamination in ground water will not exceed an acceptable level of risk at anytime over 1,000 y.

The **critical group** for individual risk is defined by NRC (1995, pp. 53–54) as

> . . . those individuals in the population who, based on cautious, but reasonable, assumptions, have the highest risk resulting from . . . releases. The group should be small enough to be relatively homogeneous with respect to diet and other aspects of behavior that affect risks. The critical group includes the individuals at maximum risk and is homogeneous with respect to risk.
>
> A group can be considered homogeneous if the distribution of individual risk within the group lies within a total range of a factor of 10, and the ratio of the

> mean of individual risks in the group to the standard (i.e., acceptable level of risk) is less than one-tenth. If the ratio of the mean group risk to the standard is greater than or equal to one, the range of risk within the group must be within a factor of 3 for the group to be considered homogeneous. For groups with ratios of mean group risk to the standard between one-tenth and one, homogeneity requires a range of risk interpolated between these limits.

The latter approach, recommended by the NRC (1995), which uses the average risk in the critical group as the basis for comparison with the acceptable level of risk, avoids using the acceptable level of risk as a standard for protecting a person with unusual habits or sensitivities. Nevertheless, by avoiding use of unreasonable assumptions concerning habits and sensitivities affecting risk, this recommended process is considered to afford a high level of protection for most persons, when the threshold for risk acceptability is not exceeded.

Risk-Based Boundary

Establishing the extent of a contaminant boundary for a CAU based on morbidity cancer risk is the most scientifically defensible rationale for a health-protective approach. Morbidity cancer risk is the most exhaustive endpoint because it addresses both fatal and nonfatal cancers, and can be computed for the range of risk identified by EPA (2000a) as reasonable for establishing regulatory standards for drinking water contaminants, including radionuclides generally (i.e., an excess lifetime cancer risk that does not exceed 10^{-4} [1/10,000] and ideally is less than 10^{-6} [1/1,000,000]). A value for risk in this range can be selected to represent a *de facto, "de minimus"* level of risk (i.e., so low it can be considered a negligible incremental risk). However, as explained by NRC (1995, p. 60), defining such a level of negligibility for risk really is a policy decision. Nevertheless, for purposes of this discussion and for illustrative purposes only, public drinking-water consumption will be used as the only exposure pathway, and limiting 70-year lifetime excess-cancer *morbidity* risk to 10^{-4} will be considered health-protective and a legitimate starting point for deriving a risk-based contaminant boundary for a CAU. In practice the health-risk calculations for the contaminant boundary will be based on the described and agreed-to land-use scenario.

Accordingly, based upon the transport model simulations, the predicted risk for any model location **x** can be computed as

$$R(\mathbf{x},t) = \sum_{i}^{n} \bar{c}_i(\mathbf{x},t) \times E \times r_i, \tag{1}$$

where

$R(\mathbf{x}, t)$ = the lifetime excess morbidity (fatal and nonfatal) cancer risk applicable to an average member of the public exposed to a constant activity concentration $\bar{c}_i(\mathbf{x},t)$ at a discrete location **x** = (x, y, z) over the 70-y lifetime beginning at time t where $0 < t \leq 1{,}000$ y;

$$\bar{c}_i(\mathbf{x},t) = \frac{1}{70\,y}\int_{t}^{t+70} c_i(\mathbf{x},t')\,dt' =$$

a 70-y average activity-based concentration (Bq/L) of radionuclide i at a discrete location **x** during a 70-y lifetime beginning at any time t, where $0 < t \leq 1{,}000$ y;

E = the exposure of 51,100 L (per 70-y lifetime) based on the generally accepted regulatory practice of using conservative values of 2 L/d for 365 d/y over the 70-y lifetime; and

r_i = the radionuclide-specific morbidity cancer risk coefficient (Sv/Bq; see Appendix A for description of units) addressing average radiogenic cancer risk, whether or not the cancer is fatal, for ingestion of drinking water (from tabulation in Table 2.2a in EPA, 1999).

Consequently, based upon the model solution $c_i(\mathbf{x},t)$, the corresponding risk solution $R(\mathbf{x},t)$ can be computed at all discrete locations and at all times of the simulation (with simulations carried out to allow for the constant activity concentration $\bar{c}_i(\mathbf{x},t)$ to be evaluated for lifetimes between 0 and 1,000 y). Hence, the risk-based standard may be considered "met" at location **x** and time t if $R(\mathbf{x},t)$ is equal to or less than 10^{-4}, while, conversely, it is considered violated or exceeded if $R(\mathbf{x},t)$ is greater than 10^{-4}.

Alternatively, to apply the concept of *expected value* of risk for a critical group, the ensemble of risk solutions, R, at each groundwater location **x** and a time step t can be averaged over the ensemble of n realizations to yield a mean value, $\overline{R}(\mathbf{x},t)$, which then can be compared with the acceptable level of risk. For the purposes of this discussion, the locus of points where the average risk for the critical group does not exceed the acceptable value of risk of 10^{-4}, at any time t, can be considered locations outside or beyond the contaminant boundary. Therefore, under such conditions, there is an expectation that groundwater from these locations would not contain radionuclides at activity concentrations that would produce unacceptable levels of risk for most persons in the critical group.

The MCL-Based Boundary and Conformity with Health Protection

As already mentioned, the currently promulgated National Primary Drinking Water Regulations (NPDWRs) for radionuclides (see EPA, 2000a; specifically CFR 40 Part 141, Subpart G, Section 141.66, *Maximum Contaminant Levels for Radionuclides*), which the 1986 reauthorization of the Safe Drinking Water Act required EPA to develop and implement, were conceived using now outdated concepts and metabolic and dosimetric models for gross alpha-emitting radionuclides (excluding radon and uranium) and for beta particle and photon radioactivity categories of radionuclides (see EPA, 1976). Additionally, the current MCLs for radionuclides, specified by the NPDWRs for radionuclides, do not focus adequately on health-risk consequences that could evolve from the, albeit unusual, situation involving the presence of multiple radionuclides in a drinking water supply, especially when different categories of radiation emitters are present simultaneously and also include the presence of uranium or radon (radon will not be dealt with in this discussion because it is not an introduced source term for UGTA). Finally, the NPDWRs overemphasize cost-effectiveness and convenience of sampling, monitoring, and detection practices, as justification for having activity- or mass-based concentration standards for three different categories of radionuclides (i.e., alpha-emitting; beta- and photon-emitting; and uranium).

For the majority of *individual* radionuclides in both the alpha and beta categories, the product of

- An individual radionuclide's MCL: 15 pCi/L established for alpha, or the respective activity concentration derived for a beta- or photon-emitting radionuclide in accordance with an annual dose-equivalent limit of 4 mrem/y to a critical organ;

- The oversimplified and conservative combination of ingestion-exposure rate and duration terms used for regulatory practice, and applied here for illustrative purposes only (i.e., 2 L/d × 365 d/y × 70 y/lifetime = 51,100 L/lifetime), and

- The most recent radionuclide-specific lifetime excess-cancer mortality and/or morbidity risk coefficient published by EPA (1999) given in units of risk/Bq (where 1 Bq = 0.037 pCi), and determined using up-to-date metabolic and dosimetric models and related modern concepts,

yields respective lifetime excess-cancer mortality or morbidity risks that are at levels in the middle to upper end of the lifetime excess-cancer risk range that is considered acceptable in regulatory practice (i.e., not to exceed 10^{-4} [1/10,000] and ideally less than 10^{-6} [1/1,000,000]), although there are a few radionuclides having risk values that exceed this range (EPA, 2000b; EPA/USGS 2000). Tables 1 and 2 are designed to illustrate this point and present selected radionuclides that are either beta-/photon- or alpha-emitting.

From Table 1 it is clear that cesium-137, at its MCL, could alone produce mortality and morbidity risk exceeding the range of acceptability. From Table 2, it is apparent that combinations of alpha-emitting radionuclides, and combinations of beta-/photon-emitting radionuclides, may be in compliance with the MCLs for those categories, and yet each individual category may have summed levels of risk that are above the more health-protective end of the target range for risk acceptability ($\leq 10^{-6}$). Even more importantly, the illustrative calculations appearing in Table 2 reveal that it is possible for categorical MCLs to be achieved when both alpha- and beta-emitting radionuclides are present, but together the combination of these alpha- and beta-emitting radionuclides can produce levels of lifetime excess cancer mortality and morbidity risk that would exceed the target-risk range considered to be health-protective in regulatory practice (i.e., 10^{-4} to $\leq 10^{-6}$). Thus, in order to be health-protective unambiguously, it seems more appropriate to construct risk-based contaminant boundaries for CAUs that do not depend strictly on MCLs defined by the NPDWRs.

Table 1. Example of calculated lifetime excess mortality and morbidity cancer risks for *selected* radionuclides at their individual regulatory limit (i.e., as if no other radionuclide were present).

Radionuclide	Symbol $\left({}^{A}_{Z}X\right)$[a] and main decay emission (category)		Half life $\left(t_{1/2};\ y\right)$	MCL (pCi/L)[b]	Cancer risk coefficients for drinking water ingestion (Risk/Bq) (from Table 2.2a in EPA, 1999)		Calculated lifetime excess cancer risk[c]	
					Mortality	Morbidity	Mortality	Morbidity
Tritium	${}^{3}_{1}H$	(β)	1.23×10^{1}	20,000	9.44×10^{-13}	1.37×10^{-12}	3.57×10^{-5}	5.18×10^{-5}
Strontium-90	${}^{90}_{38}Sr$	(β)	2.91×10^{1}	8	1.34×10^{-9}	1.51×10^{-9}	2.03×10^{-5}	2.28×10^{-5}
Iodine-129[d]	${}^{129}_{53}I$	(β)	1.57×10^{7}	1	4.07×10^{-10}	3.99×10^{-9}	7.70×10^{-7}	7.54×10^{-6}
Cesium-137[d]	${}^{137}_{55}Cs$	(β)	3.02×10^{1}	200	5.66×10^{-10}	8.22×10^{-10}	2.14×10^{-4}	3.11×10^{-4}
Plutonium-238[d]	${}^{238}_{94}Pu$	(α)	8.77×10^{1}	15	2.75×10^{-9}	3.55×10^{-9}	7.80×10^{-5}	1.01×10^{-4}
Plutonium-239[d]	${}^{239}_{94}Pu$	(α)	2.41×10^{4}	15	2.85×10^{-9}	3.64×10^{-9}	8.08×10^{-5}	1.03×10^{-4}
Americium-241	${}^{241}_{95}Am$	(α)	4.33×10^{2}	15	2.01×10^{-9}	2.81×10^{-9}	5.70×10^{-5}	7.97×10^{-5}

[a] Isotopes are nuclides of an element (X), with the same number of protons (p), but a different number of neutrons (n) in the nucleus. The atomic mass number A (= n + p), and atomic number Z (= p) are shown for the selected radionuclides listed from beta (β) and alpha (a) emission categories.

[b] Maximum contaminant level (MCL) for radionuclides present by themselves: derived for the beta-particle emitters based on an annual dose-equivalent limit of 4 mrem/y (see EPA, 1976; 2000a,b), and specified for the alpha particle emitters (except radon and uranium) as not to exceed a total of 15 pCi/L (see EPA, 2000a). Radionuclides of naturally occurring uranium (U-234, U-235, and U-238) are alpha-emitting; however, uranium is regulated at 30 mg/L to prevent kidney toxicity. To convert *pCi to Bq*, multiply by 0.037 Bq/pCi.

[c] Lifetime excess cancer risk calculation: MCL (pCi/L) × 0.037 (Bq/pCi) × [2 (L/d) × 365 (d/y) × 70 (y/lifetime)] × cancer risk coefficient (Risk/Bq). Morbidity cancer risk is the most exhaustive endpoint as it addresses both fatal and nonfatal cancers.

[d] *Note:* Except for iodine-129, the other radionuclides associated with this footnote, when present at their MCLs, are in the middle or very near the highest level of the risk range that is considered acceptable (i.e., 10^{-4} to 10^{-6}); and in the case of cesium-137 mortality and morbidity cancer risks, and plutonium-238 and plutonium-239 morbidity risk the values even exceed the highest level of the risk range that is generally considered acceptable (i.e., 10^{-4}). The EPA (2000a) did estimate the radiogenic cancer risk for a 30 pCi/L equivalent activity of uranium (based on a mass:activity ratio of about 1:1) to be about 10^{-4} (i.e., at upper end of target-risk range).

Table 2. Example of calculated lifetime excess mortality and morbidity cancer risks for *selected* radionuclides at their *combined* maximum contaminant levels (MCLs) for gross-alpha (excluding U and Rn) and gross-beta activities (see EPA, 2000a).

Radionuclide	Symbol $\left({}^{A}_{Z}X\right)$[a] and main decay emission (category)		Individual MCL for alpha and beta emitter (pCi/L)	Emission category "adjusted" MCL for multiple radionuclides		Cancer risk coefficients for drinking water ingestion (Risk/Bq) (from Table 2.2a in EPA, 1999)		Calculated lifetime excess cancer risk[b]	
				Adjustment factor[c]	MCL (pCi/L)	Mortality	Morbidity	Mortality	Morbidity
Tritium	${}^{3}_{1}H$	(β)	20,000	0.25	5,000	9.44×10^{-13}	1.37×10^{-12}	8.92×10^{-6}	1.30×10^{-5}
Strontium-90	${}^{90}_{38}Sr$	(β)	8	0.25	2	1.34×10^{-9}	1.51×10^{-9}	5.07×10^{-6}	5.71×10^{-6}
Iodine-129	${}^{129}_{53}I$	(β)	1	0.25	0.25	4.07×10^{-10}	3.99×10^{-9}	1.92×10^{-7}	1.89×10^{-6}
Cesium-137	${}^{137}_{55}Cs$	(β)	200	0.25	50	5.66×10^{-10}	8.22×10^{-10}	5.35×10^{-5}	7.77×10^{-5}
	β-particle total			1.0 ⇒	4 mrem/y	Lifetime excess cancer risk for (β)		6.77×10^{-5}	9.83×10^{-5}
Plutonium-238	${}^{238}_{94}Pu$	(α)	15	0.33	5	2.75×10^{-9}	3.55×10^{-9}	2.60×10^{-5}	3.36×10^{-5}
Plutonium-239	${}^{239}_{94}Pu$	(α)	15	0.33	5	2.85×10^{-9}	3.64×10^{-9}	2.69×10^{-5}	3.44×10^{-5}
Americium-241	${}^{241}_{95}Am$	(α)	15	0.33	5	2.01×10^{-9}	2.81×10^{-9}	1.90×10^{-5}	2.66×10^{-5}
	α-particle total			1.0 ⇒	15	Lifetime excess cancer risk for (α)		7.19×10^{-5}	9.45×10^{-5}
	Total lifetime excess cancer risk for both β and α particle categories							***1.40×10^{-4}***	***1.93×10^{-4}***

[a] Isotopes are nuclides of an element (X), with the same number of protons (p), but a different number of neutrons (n) in the nucleus. The atomic mass number A (= n + p), and atomic number Z (= p) are shown for the selected radionuclides listed from beta (β).and alpha (a; excluding uranium and radon) emission categories.

[b] Lifetime excess cancer risk calculation: MCL (pCi/L) × 0.037 (Bq/pCi) × [2 (L/d) × 365 (d/y) × 70 (y/lifetime)] × cancer risk coefficient (Risk/Bq). Morbidity cancer risk is the most exhaustive endpoint as it addresses both fatal and nonfatal cancers.

[c] For one or more β- or one or more a-emitting radionuclides the total-dose and activity-concentration limit must be ≤4 mrem/y or ≤15 pCi/L, respectively. Therefore, when multiple radionuclides are present, each MCL is divided by the number of radionuclides present so that the contributions from all β-emitting radionuclides can equal 4 mrem/y and the contributions from all a-emitting radionuclides can equal 15 pCi/L.

Nevertheless, to honor the language of the FFACO (1996; as amended) it may be important to consider development of contaminant boundaries that are based on the current MCLs for the radionuclides of concern (EPA, 2000a), as well as an acceptable level of risk (which in practice would be a function of exposure[s] resulting from a realistic assumption of future land use). Recall that although the MCLs are currently enforceable, they are categorical in nature and do not directly address risk.

The procedure for computing a MCL-based contaminant boundary involves the following iterative process. First, the hydrologic-flow and contaminant-transport model predictions for concentrations of each radionuclide at every location and in each year of interest between 0 and 1,000 y will be generated and saved. Note that because averaging in the sense of Eq. 1 is not used in the MCL approach, only 1,000 years of simulations are required. Also, recall that because of uncertainties in the actual application, a multiple number of simulation results will be developed in this fashion. These annual activity-concentration values will then be separated and converted into appropriate categories and units for comparison with current MCLs (α, which is activity-based; β, which is dose-based; and uranium, which is mass-based) described by the NPDWRs (see note c in Table 2). Then, for every simulation run at every location in every year, agreement with the MCLs for the radionuclides of concern will be determined.

For each year and at each model location, or grid block, the fraction of the ensemble simulation results that exceed any one or all of the currently applicable categorical MCLs for the respective radionuclides of concern (i.e., EPA, 2000a: gross-alpha activity concentration [excluding uranium and radon] not to exceed 15 pCi/L [to convert to Bq/L multiply by 0.037 Bq/pCi]; gross-beta/photon activity concentrations not to yield an annual dose equivalent in excess of 4 mrem/y to the total body or any internal organ; and uranium mass concentration not to exceed 30 μg/L) will be determined. Where, for purposes of this example, 95% or more of the simulations for a particular location **x** in the modeling domain reveal that none of the MCLs standards are exceeded (either at a particular time [t] or over in the entire 1,000-y period of concern), such locations will be considered **external** to the contaminant boundary. (Note, it is not considered appropriate to apply an expected value method here, as that procedure was recommended by NRC [1995] to be applied to *risk* for a critical group).

Next, the two-dimensional surface projection of the perimeter describing this MCL-based contaminant boundary will be compared with the two-dimensional surface projection describing a risk-based contaminant boundary (which in practice is based on agreement about realistic future land use) with an equal level of confidence. Again, for purposes of this discussion, consider the risk-based contaminant boundary that is used for comparison to be determined based on those locations of groundwater that are unlikely to produce an unacceptable level of risk (i.e., for purposes of this example, only, a value $>10^{-4}$). Thus, if the MCL-based contaminant boundary conforms to or is inside of the risk-based contaminant boundary, then groundwater at locations at or inside the MCL-based contaminant boundary may exceed both acceptable risk and the acceptable MCLs, and at locations in between an MCL- and risk-based contour, MCLs are unlikely to exceed acceptable levels, even though risk may exceed acceptable levels.

Alternatively, if the contour of the MCL-based contaminant boundary is outside of the risk-based contaminant boundary, then beyond both boundaries the groundwater is unlikely to contain radionuclides at activity concentrations that produce unacceptable levels of risk or that exceed MCLs, and in between the risk-based contaminant boundary and the MCL-based contaminant boundary (that extends beyond it), MCLs may be exceeded. However, at such locations the radionuclides in groundwater are likely to yield risk that is less than or equal to an acceptable level. Consequently, if both MCL-based and risk-based levels of acceptability are to be achieved, and risk is unambiguous with respect to protecting public health, then in this case the contaminant boundary is either where the two contours are exactly the same, or where the MCL-based contour is beyond the risk-based contour. Thus, beyond the MCL-based contour the radionuclides in groundwater are unlikely to exceed the MCL or produce an unacceptable level of risk. Nevertheless, in this example, the risk-based approach seems by itself to be sufficiently health-protective.

Fostering Risk Communication to Educate, Inform, Enlighten, and Fully Disclose the Goals and Structure of Contaminant Boundaries for CAUs at the NTS

As Moeller (1992, see p. 227) stated in his book, *Environmental Health*:

> One fundamental problem that should be addressed . . . is the common public misconception that standards assure risk-free conditions.

Furthermore, he observes astutely:

> Experience has shown that, although the public will voluntarily accept risks, they will balk at accepting involuntary risks, particularly if they conclude that the risks are being imposed on them without full disclosure.

For these two reasons, a continuing dialogue among scientists, decision makers, stakeholders, and the interested public will be crucial, if scientifically defensible and timely risk-management and risk-reduction actions are to be executed (e.g., establishing compliance boundaries) for CAUs. Pursuing and fostering this dialogue will help to correct misconceptions, improve understanding, and redirect perceptions from the sensational and emotional to the scientifically factual, especially when involuntary risks are considered to be imposed. In fact, it is important to mention in such discussions that if groundwater internal (or even external) to a contaminant boundary were not consumed or used at all (e.g., due to denial of use; or a lack of accessibility), regardless of the level of contamination, there would be zero risk and no potential health consequences to consider, because there would be no exposure.

The importance of communication cannot be overemphasized because scientific data can be highly technical, subject to interpretation, and often difficult to explain. Furthermore, numerical and even analytical computer models used today to study complex environmental processes can be filled with nuance and intricacy, and of course, both model and parametric uncertainties that can be difficult to quantify and sometimes even hard to qualify and describe effectively. Nevertheless, a continuing engagement and exchange of information between all parties in an environment of mutual respect can help to address exaggerated expectations for regulations or technology, explain the adoption of plausible conservatisms, and even describe the fundamental concepts related to numerical and analytical models and modeling and the nature of the probabilistic results that are typically generated. Executing risk communication as a long-term strategy can ensure credible cost-effective, practical, and satisfactory solutions can be

reached in a timely manner for the long-term protection of public health and particularly with regard to radionuclide contamination of groundwater.

Conclusion

Evidence is provided in this document that explains the goals and methodology for constructing risk-based contaminant area boundaries for a CAU at the NTS. An approach is also suggested for deriving such boundaries so that they would also comply with the current MCLs established by the NPDWRs for the radionuclides of concern.

The described methodology is similar but not identical to the approach employed by Maxwell et al. (1998) for addressing a contaminant source, its mobility in groundwater, subsequent exposure, and health consequences quantitatively, and has more fidelity than the approach Daniels et al. (1993) and Andricevic et al. (1994) used for evaluating health consequences quantitatively from ingestion exposure of radionuclide-contaminated groundwater at the location of a nuclear test. Further, the variability and uncertainty of exposure parameters can and should be addressed more quantitatively, as was done most recently by Daniels et al. (2000), and even quantifying nominal uncertainty by the procedure described by EPA (1999) for each of the radionuclide-specific morbidity cancer risk coefficients that are used should also be considered.

Additionally, through risk communication a more informed, reasonable, and cooperative decision making process will emerge between all members of the participating community—scientists, decision makers, stakeholders, and the interested public. This will result in risk-management decisions that are timely, effective, collaborative, and informed. A well-executed risk communication effort with respect to the goals and structure of the contaminant boundaries for a CAU will provide a framework for developing a mutually agreeable compliance boundary that is health-protective and scientifically reasonable while meeting regulatory requirements and institutional agreements, and that also is practical, realistic, and economically feasible.

Acknowledgments

This document was prepared as a technical product of the Underground Test Area Technical Working Group (UGTA-TWG) and constitutes work performed under the auspices of the U.S. Department of Energy by Lawrence Livermore National Laboratory under Contract W-7405-Eng-48. Gratitude is extended to the members of the UGTA-TWG *Ad Hoc* Subcommittee for Risk Assessment: J. Daniels (Chair; LLNL); B. Deshler (Shaw, E&I, Inc.); R. Jacobson (DRI); M. Sully (former employee of GeoTrans, Inc.); and A. Tompson (LLNL) for their suggestions, recommendations, and technical assistance, as well as to Ahmed Hassan (DRI) for his insight. Appreciation also is extended to K. Rehfeldt (Geotrans, Inc.) and G. Pohll (DRI) for their technical consideration and recommendations. Additionally, we thank R. Bangerter, the UGTA Project Manager, located at the U.S. Department of Energy (DOE), National Nuclear Security Administration Nevada Site Office (NNSA/NSO) in Las Vegas, NV, for his constructive comments, which improved the focus and quality of this document. Finally, the review performed by members of the Division of Environmental Protection (NDEP) at the State of Nevada Department of Conservation and Natural Resources was quite beneficial and exceptionally valuable.

Note that the views and opinions of the authors expressed herein do not necessarily state or reflect at this time those of the United States Government or the University of California, and shall not be used for advertising or product endorsement purposes.

References

Andricevic, R., J.I. Daniels, and R.L. Jacobson (1994), "Radionuclide Migration Using a Travel Time Transport Approach and Its Application in Risk Analysis," *J. Hydrol.* **163***,* 125–145.

Daniels, J.I., K.T. Bogen, and L.C. Hall (2000), "Analysis of Uncertainty and Variability in Exposure to Characterize Risk: Case Study Involving Trichlorethylene Groundwater Contamination at Beale Air Force Base in California," *Water, Air, Soil Pollut.* **123***,* 273–298.

Daniels, J.I., R. Andricevic, L.R. Anspaugh, and R.L. Jacobson (1993), "4. Risk-Based Screening Analysis of Ground Water Contaminated by Radionuclides Introduced at the Nevada Test Site," in *Pilot Study Risk Assessment for Selected Problems at the Nevada Test Site (NTS),* J.I. Daniels, Ed., Lawrence Livermore National Laboratory, Livermore, CA, UCRL-LR-113891.

Federal Facility Agreement and Consent Order (FFACO) (1996; as amended), *Appendix VI. Corrective Action Strategy Section 3. Underground Test Area,* Agreed to by the State of Nevada, the U.S. Department of nergy, and the U.S. Department of Defense.

Maxwell, R.M., S.D. Pelmulder, A.F.B. Tompson, and W.E. Kastenberg (1998), "On the Development of a New Methodology for Groundwater-Driven Health Risk Assessment," *Water Resour. Res.* **34***,* 833–847.

Moeller, D.W. (1992), "Chapter 10. Standards," in *Environmental Health* (Harvard University Press, Cambridge, MA), pp. 225–228.

National Research Council (NRC) Committee on Technical Bases for Yucca Mountain Standards (1995), *Technical Bases for Yucca Mountain Standards,* National Academy Press, Washington, DC.

Pawloski, G.A., A.F.B. Tompson, and S.F. Carle, Eds. (2001), "Appendix A: Radiological Source Term," in *Evaluation of the Hydrologic Source term from Underground Nuclear Tests on Pahute Mesa at the Nevada Test Site: The CHESHIRE Test,* Lawrence Livermore National Laboratory, Livermore, CA, UCRL-ID-147023 (see especially Table A-1, pp. A-5 to A-8).

United States Environmental Protection Agency (EPA) (1976), *National Interim Primary Drinking Water Regulations,* U.S. Environmental Protection Agency, Office of Water supply, Washington, DC, EPA-570-9-76-003; Reproduced by U.S. Department of Commerce National Technical Information Service, Springfield, VA; NTIS Accession No. PB267630 (1976); see *Appendix B–Radionuclides* (pp. 129–142) and *Appendix IV (in Appendix B): Dosimetric Calculations for Man-Made Radioactivity,* (pp. 152–159).

United States Environmental Protection Agency (EPA) (1988), *Federal Guidance Report No. 11: Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion,* Office of Radiation Programs, United States Environmental Protection Agency, Washington, DC, EPA-520/1-88-020 (September 1988).

United States Environmental Protection Agency (EPA) (1999), *Federal Guidance Report No. 13: Cancer Risk Coefficients for Environmental Exposure to Radionuclides,* Office of Radiation and Indoor Air, United States Environmental Protection Agency, Washington, DC, EPA 402-R-99-001.

United States Environmental Protection Agency (EPA) (2000a), "Part II. Environmental Protection Agency, 40 CFR Parts 9, 141, and 142: National Primary Drinking Water Regulations; Radionuclides; Final Rule," *Fed. Regist. 65*(236), 76708–76753 (December 7, 2000).

United States Environmental Protection Agency (EPA) (2000b), "Part IV. Environmental Protection Agency, 40 CFR Parts 9, 141, and 142: National Primary Drinking Water Regulations; Radionuclides; Notice of Data Availability; Proposed Rule," *Fed. Regist. 65*(78), 21576–21628 (April 21, 2000), see Table II-3, p. 21605–21614.

United States Environmental Protection Agency, Office of Ground Water and Drinking Water, in collaboration with United State Environmental Protection Agency, Office of Indoor Air and Radiation, and United States Geological Survey (EPA/USGS) (2000), *Radionuclides Notice of Data Availability, Technical Support Document,* Targeting and Analysis Branch, Standards and Risk Management Division, Office of Ground Water and Drinking Water, United States Environmental Protection Agency, Washington, DC (March 2000), see Tables III-3 and III-4, pp. III-20 to III-29 (http://www.epa.gov/safewater/standard/pp/radnucpp.html#noda; accessible September 2003).

Appendix A:
Radiation, Radioactivity, Related Dose and Its Special Units, and Risk Coefficients and Their Applications

Ionizing radiation is photon (including high-frequency UV; X and gamma rays, and cosmic radiation) and alpha- and beta-particle emissions of sufficiently high energy (e.g., ≥34 eV) that they interact with the atoms composing matter, stripping orbital electrons, and creating ion pairs. When ionizing radiation penetrates cells it deposits energy and directly ionizes biological materials in this manner, and it can also create toxic agents from the ionization of cellular water molecules, such as free radicals and hydrogen peroxide. Even at low doses, radiation is considered to pose some likelihood of lifetime excess cancer risk (i.e., a stochastic effect), which can be quantitatively equated to either morbidity (defined as fatal and nonfatal cancers) or mortality (defined as fatal cancers only).

The activity of a radioactive isotope (i.e., radionuclide) is defined as the number of atomic transformations per second. One atom transformed per second is equal to 1 Becquerel (Bq, in International System of Units, SI), and 3.7×10^{10} Bq equates to 1 Curie (Ci, the special unit of activity used before SI units were adopted). However, a Bq of activity does not necessarily equate to the number of radiations emitted by the radioactive isotope as it decays. For example, tritium (^{3}H) releases 1 β particle per atom transformed (i.e., 1 radiation/s) but cobalt-60 (^{60}Co) releases 1 β particle and 2 γ rays per atom transformed (i.e., 3 radiations/s), and potassium-42 (^{42}K) releases 1 β particle and 0.2 γ rays per atom transformed (i.e., 1.2 radiations/s).

Because activity measures the quantity of atomic transformations per second for a radionuclide, and atomic transformations release ionizing radiation, it is the activity and *not the mass* of a radioactive isotope that is relevant for assessing health risk. Yet, modeling groundwater flow and radionuclide transport involves accounting for the mass of a radionuclide per unit volume of water (concentration). However, the mass of a radionuclide of concern per unit volume of water can be converted to its activity per unit volume of water (i.e., activity concentration) using the specific activity (*SA*) for that radionuclide.

Technically, the specific activity (*SA*) is the relationship between mass (mole or g) and activity (Bq) for a radionuclide, such that $SA = \lambda N$ [Bq/mole (or Bq/g)], where the decay

constant (λ) is expressed as 1/seconds, and equals $0.693/t_{1/2}$, and $t_{1/2}$ is the half life of the radionuclide (and conversely $t_{1/2} = 0.693/\lambda$), which is the time for one-half of the parent atoms that are present for a radionuclide to decay to daughter products (progeny), and N = atoms/mole (or atoms/g). Thus, the moles or grams of a radionuclide can be determined from its activity and visa versa using the relationship shown in Eq. A.1:

$$\text{Mole (or g)} = \frac{\text{Bq (atoms transformed/s)}}{\left[\frac{0.693}{t_{1/2}(\text{s})} \times \frac{6.023 \times 10^{23}\ \text{atoms}}{\text{mole (or g)}}\right]} . \qquad \text{(A.1)}$$

Accordingly, knowing only the mass per unit volume in water (e.g., μg/L) and the half life (i.e., $t_{1/2}$ in seconds) for a radionuclide, the relationship shown in Eq. A.1 can be used to determine the corresponding activity concentration (Bq/L) from the mass concentration (e.g., μg/L), and that value can be used to evaluate health effects from exposure to the radiation emitted by the radionuclides of concern in the water.

The *absorbed dose* from exposure to ionizing radiation is expressed in units of gray (Gy, in SI units), where 1 Gy is defined as the energy per unit mass imparted to matter equal to 1 J/kg (and it is also equal to 100 rad, where rad is the International Commission on Radiation Units and Measurements [ICRU] special dimension used before SI units were adopted). The *dose equivalent* is the product of the absorbed dose and a dimensionless radiation quality or weighting factor (Q, representing an adjustment factor for distinguishing between the biological effectiveness of different types of radiation; for example, Q = 1 for beta, gamma, and X-ray emissions, and Q = 20 for alpha emissions) and is expressed in units of sievert (Sv), where 1 Sv represents the energy per unit mass imparted to biological material equal to 1 J/kg (and it is also equal to 100 rem, where rem is the ICRU special dimension used before SI units were adopted). The *effective dose equivalent (ede)* is a concept for converting internal exposures to an equivalent whole-body dose (by summing the products of tissue-weighting factors and equivalent doses to an organ) so that doses from internal exposures can be added to those from external exposures to determine a total dose (see Moeller, 1992). The *ede* also is expressed in units of Sv, and the tissue-weighting factor represents the fractional contribution that an individual body organ can make, as a consequence of internal exposure to radiation, to the total

lifetime excess cancer risk were the entire body (i.e., all organs) to be irradiated uniformly (EPA, 1988).

To account for dose that is being delivered to tissue(s) as long as a radionuclide is present inside the body and decaying, the *committed effective dose equivalent (cede)* was conceived. This derived value is also expressed in Sv, is defined as the sum of all doses projected to be received in the future from the intake of radionuclides in the current year, and by convention, the period over which a dose is considered committed in a tissue is 50 y following intake, a value originally selected arbitrarily for application to occupational exposures (EPA, 1988).

The *cede* agrees notionally with modern metabolic and dosimetric understanding, and the *cede* per unit intake (Sv/Bq), or *cede* applicable "dose conversion factor," for internal exposure by ingestion, has been calculated and published by EPA (1988) and can be used to establish intake and concentration guidance in a regulatory context. Such application is considered appropriate because for radionuclides with long *effective half lives* (i.e., long radioactive half lives in combination with a long residence time in the body, especially relative to a 50-y period of commitment or even a 70-y lifetime) the *cede* may overestimate the total dose that actually would be expected to occur over the 50-y commitment period (Moeller, 1992), or by implication, even a 70-y lifetime.

In contrast with using the *cede* approach for deriving intake and concentration limits for radionuclides, there is another outdated method that is not as valid, but has regulatory precedent. For example, the National Primary Drinking Water Regulations for radionuclides (EPA 2000a) identify 0.04 mSv/y (4 mrem/y) as the total annual dose equivalent to an organ or the whole body that cannot be exceeded from internal exposure to beta particle and photon radioactivity, and then this annual dose equivalent, along with a 2 L/d consumption rate, is used as a basis for setting annual average activity-concentration limits for drinking water intake for beta-particle- and photon-emitting radionuclides (see EPA 1976 for procedure). This method is best described as the "critical-organ dose-limit approach" because it simply involves using an acceptable dose limit for a critical organ as a fundamental parameter for setting a concentration limit.

Today, a far superior and more defensible procedure exists as an alternative to using the "critical-organ dose-limit approach," or even applying *cede* dose conversion factors, for developing concentration limits for radionuclides. This new procedure capitalizes on the

development of radionuclide-specific lifetime radiogenic cancer risk coefficients (expressed as either a cancer mortality [fatal only] risk per unit activity or as a cancer morbidity [fatal and nonfatal combined] risk per unit activity [i.e., mortality risk/Bq or morbidity risk/Bq]) for the U.S. population published by EPA (1999), and derived using

> . . . state-of-the-art methods and models that take into account age and gender dependence of intake, metabolism, dosimetry, radiogenic risk, and competing causes of death in estimating the risks to health from internal or external exposure to radionuclides.

Furthermore, according to EPA (1999), the purpose of these cancer risk coefficients is intended

> . . . to support rulemaking . . . and . . . encouraged to promote consistency in risk assessment . . . .

especially by Federal agencies.

In fact, the published cancer risk coefficients are suited for use in *prospective assessments* of potential cancer risk related to long-term exposure to radionuclides in environmental media. Specifically, they have been tabulated for over 800 radionuclides and include values applicable to low-acute doses or low-dose rates from internal exposure through various media, including drinking water. Technically, these radionuclide-specific cancer-risk coefficients can be applied to estimate the lifetime excess mortality or morbidity cancer risk due to chronic exposure over a lifetime to a constant environmental activity concentration by an average individual in a stationary population in the U.S. Consequently, summing the products of drinking water activity concentrations predicted for specific radionuclides (Bq/L), a corresponding radionuclide-specific cancer risk coefficient (e.g., to be exhaustive, cancer morbidity risk/Bq), and a conservative estimation of a lifetime exposure to drinking water (e.g., 2 L/d × 365 d/y × 70 y/lifetime), will yield a total lifetime excess cancer (e.g., morbidity, to be exhaustive) risk for exposure to all of the radionuclides considered.

The computed total lifetime excess cancer (e.g., morbidity, to be exhaustive) risk can then be compared to a *de facto,* "*de minimus*" level of risk (i.e., so low it can be considered negligible) that can be selected from the within the range of risk identified by EPA as reasonable for establishing regulatory standards for drinking water contaminants, including radionuclides generally (i.e., an excess lifetime cancer risk that does not exceed 10^{-4} [1/10,000] and ideally is less than 10^{-6} [1/1,000,000]). For example, limiting lifetime excess cancer morbidity risk to 10^{-4}

or less would be a legitimate unifying consideration with regard to establishing a reasonably well conceived health-protective contaminant boundary perimeter for a CAU.

Finally, the computational model used to generate the mortality and morbidity risk coefficients for the 800 radionuclides is a relatively complex formulation with numerous parameters that depend on time, age, and gender, which make a systematic quantitative uncertainty analysis an overwhelming and difficult task that was not performed (EPA, 1999). For this reason, and because a full-scale parameter uncertainty analysis applicable to a simpler model that produces results in reasonable agreement with those generated by the more complex model, also is not considered feasible for implementation for a large number of radionuclides (EPA, 1999), the risk coefficients are taken as a group to be centrally located within their uncertainty range, even though this may not always be true for some individual radionuclide-specific cancer risk coefficients (EPA, 1999). Furthermore, application of the published tabulated individual radionuclide-specific cancer risk coefficients would likely conform to accepted regulatory practice and produce results that would be considered sufficient pending additional review. Therefore, the use of the published radionuclide-specific cancer risk coefficients without addressing in detail the quantification of uncertainty in the estimation of any particular value is considered practical. Nevertheless, it is important to note that a nominal uncertainty value (intended to reflect major uncertainties that are largely independent of the exposure scenario) may be reasonable and useful to derive in the future for each radionuclide-specific cancer-risk coefficient that would be used, and that decision could be addressed following a critical review of the results obtained using the published values explicitly.

References for Appendix A

Moeller, D.W. (1992), "Chapter 10. Standards," in *Environmental Health* (Harvard University Press, Cambridge, MA), pp. 225–228.

United States Environmental Protection Agency (EPA) (1976), *National Interim Primary Drinking Water Regulations,* U.S. Environmental Protection Agency, Office of Water supply, Washington, DC, EPA-570-9-76-003; Reproduced by U.S. Department of Commerce National Technical Information Service, Springfield, VA; NTIS Accession No. PB267630 (1976); see *Appendix B–Radionuclides* (pp. 129–142) and *Appendix IV (in Appendix B): Dosimetric Calculations for Man-Made Radioactivity,* (pp. 152–159).

United States Environmental Protection Agency (EPA) (1988), *Federal Guidance Report No. 11: Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion,* Office of Radiation Programs, United States Environmental Protection Agency, Washington, DC, EPA-520/1-88-020 (September 1988).

United States Environmental Protection Agency (EPA) (1999), *Federal Guidance Report No. 13: Cancer Risk Coefficients for Environmental Exposure to Radionuclides,* Office of Radiation and Indoor Air, United States Environmental Protection Agency, Washington, DC, EPA 402-R-99-001.

United States Environmental Protection Agency (EPA) (2000), "Part II. Environmental Protection Agency, 40 CFR Parts 9, 141, and 142: National Primary Drinking Water Regulations; Radionuclides; Final Rule," *Fed. Regist. 65*(236), 76708–76753 (December 7, 2000).